

Report for 2003NY23B: Evaluating the Flow and Treatment of Contaminants in Urban Storm Water Infiltration Basins

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Report Follows

Evaluating the Flow and Treatment of Contaminants in Urban Storm Water Infiltration Basins

Abstract: A critical issue in designing infiltration basins is to maintain high hydraulic conductivities but to also incorporate effective pollutant removal to minimize contamination of groundwater. At an infiltration basin constructed in East New York, Brooklyn, New York, a 0.6 meter organic matter layer overlays a high-permeability rubble substrate. This research assesses the ability of the organic matter layer to remove dissolved copper (Cu), zinc (Zn), and lead (Pb) in a synthetic urban stormwater mixture applied at hydraulic application rates of 8 cm/hr to 247 cm/hr. Influent concentrations of 0.021 ppm, 0.059 ppm, and 0.28 ppm were used for Cu, Pb, and Zn, respectively. Application rates and metal concentrations are representative of actual field operating conditions. Experiments were run in replicate on three undisturbed soil columns excavated from the East New York Site. Few other studies have used undisturbed soil columns.

For all three metals, no trend was observed between removal efficiency and hydraulic application rate. The breakthrough of a chloride tracer indicates that approximately 10% of downward flow, even at low hydraulic application rates, is preferential flow. Lead was reduced to below detectable limits while Zn was consistently reduced by 70 to 80%. Cu removal rates varied more widely but were generally above 50%.

1. Introduction

Infiltration Basin History

Since the 1970's, two changes in federal regulations have placed heightened emphasis on managing and treating storm water runoff. First, in cities with combined stormwater and sanitary sewers, measures to treat the combined sewage outflow have been required. These measures can take the form of increased treatment plant capacity, expanded stormwater storage, or reductions in stormwater ever reaching the sewers. Secondly, as point source polluters of waters have decreased, an increased emphasis has been placed on controlling non-point source pollution, particularly in areas without stormwater sewers. These regulatory requirements have led to the development, design, and implementation of stormwater management practices such as detention basins, constructed wetlands, and infiltration basins. With an infiltration basin, water is collected from impervious areas and directed to the groundwater instead of readily conveyed to surface waters.

A distinct feature of infiltration basins is the high hydraulic loading rate to groundwater at the infiltration basin site due to accumulation and concentration of water. For the site in Brooklyn, the intent was to infiltrate water collected from an 8-hectare (20 acre) drainage area on a 0.4-hectare (1-acre) parcel. A one-inch storm event (return period of less than a year) would require that 20 inches be infiltrated in the basin. Measurements of infiltration rates at three typical basins on Long Island at Westbury, Syosset, and Deer Park found average infiltration rates of 27 cm/hr, 24 cm/hr, and 6 cm/hr, respectively (Seaburn and Aronson, 1974).

Due to the necessary high infiltration rates and required, sustained high hydraulic conductivities, infiltration basins have traditionally been used in areas with sandy soils. While stormwater is quickly passed to the subsurface, the rapid movement of the pollutant-laden water may result in little pollutant removal. While the use of infiltration basins in places such as Long Island, New York has not been directly linked to the deterioration of groundwater quality in the region, the presence of urban land is clearly related to the frequency of detection and concentration of compounds such as nitrate, pesticides, and volatile organic compounds (Ayers et al.,2000). A critical issue in designing infiltration basins is to maintain high hydraulic conductivities, but to also incorporate effective pollutant removal.

Characterization of Stormwater Pollutant Loading

The National Urban Runoff Program (NURP) was one of the first studies to quantify a wide range of organic and metallic toxicants in urban storm water runoff although it primarily focused on residential areas and not commercial land use (EPA, 1983). More recent work has quantified 39 different pollutants grouped in the general categories of pesticides, phthalate esters, polynuclear aromatic hydrocarbons, and metals for 9 different land use types (Pitt et al., 1995). Of approximately 50 samples, metals were detected in greater than 90% of the samples while other pollutant types were detected in less than 20% of the samples. Judging from the relative ubiquity of metals in stormwater in comparison to other pollutants, these will be the primary pollutant of interest in these experiments. Other experiments assessing the effectiveness of infiltration practices to remove pollutants have also focused on metals, primarily copper, (Cu), lead (Pb), and zinc (Zn) (Sansalone, 1999, Davis et al., 2001, Farm, 2002.). Metal concentrations are detailed in the methodology section. The synthetic stormwater mix was prepared following the guidance of Davis et al (2001). Metal concentrations are of the same order of magnitude as reported in other studies.

Existing Studies on Pollutant Removal in Infiltration Basins

Davis et al. (2001) constructed a laboratory scale bioretention basin and measured the degree of nutrient and metal removal at different depths within the system. A synthetic runoff solution was applied at a rate of 4.1 cm/hr for 6 hours. Ponding reached a maximum of 12 to 18 inches and water remained ponded for approximately two days, thus suggesting soil hydraulic conductivity was much less than the hydraulic application rate. Percent reduction of Cu, Pb, and Zn was greater than 97%.

Sansalone (1999) conducted column experiments to assess the metal removal efficiency of oxide coated sands. Influent was applied at a rate of 263 cm/hr until breakthrough occurred. For a column incorporating 90-mm of porous pavement aggregate and 522-mm of oxide coated sand, no Pb or Cu breakthrough was observed for >1000 pore volumes while Zn breakthrough was observed after 200 pore volumes. To facilitate more rapid breakthrough, influent Cu and Pb concentrations were approximately 100 times concentrations found in actual stormwater while the influent Zn concentration was approximately 10 times typical concentrations.

Farm (2002) assessed the capacity of peat, zeolite, and calcium silicate rock (opoka) to remove metals. Using metal salt concentrations of between 0.5 and 0.1 mg/L (same order of magnitude as Davis et al.), Farm(2002) considered metal removal capacity at application rates varying from 100 cm/hr to 500 cm/hr. For a 28-mm column consisting of a mixture of peat and opoka, nearly 100% of Zn was removed for hydraulic loads up to 500 cm/hr while upwards of 80% of Cu was removed at similar hydraulic loads. Assessing other substrate materials, Farm found that metal removal efficiency generally declined with increasing flow rate although this was not observed for the opoka/peat mixture. However, Farm reported that clogging of the column bottom occurred suggesting that flow rates through the column may not have been proportional to apparent application rates.

While most studies have suggested that the Cu removal efficiency is relatively similar to that of Zn and Pb, there is one case of where very low Cu removal has been reported. A composted-leaf filter developed as a compact, manufactured treatment device had reported removal efficiencies of 83% for zinc and lead but only 7% for copper (CSF Treatment Systems).

Based on these previous studies, high metal removal efficiencies are expected at the highest hydraulic loading rates anticipated in real world infiltration basins. However, as these experiments all made use of disturbed columns packed within a laboratory setting, no previous work has made use of undisturbed soil columns to study the impact of preferential flow on removal efficiencies.

2. Methodology

Site Background

The Gaia Institute in cooperation with the New York City Department of Parks and Department of Environmental Protection initiated constructed of an infiltration gallery/wetland in a vacant plot of land in East New York, Borough of Brooklyn, New York in 2001. The infiltration gallery/ wetland was intended to demonstrate the feasibility of employing underused parcels of land for managing storm water runoff and providing urban green space.

The site is located on a filled wetland, formerly part of the Jamaica Bay estuary. While the exact subsurface characteristics of this site are not known, approximately 3 to 5 meters (10 to 15 feet) of construction debris, rubble, and other landfill material were used to raise the ground surface elevation above its original wetland level sometime in the early 20th century. To construct the infiltration gallery/wetland, approximately 0.6 meters (2 feet) of the fill material was removed and replaced with an organic compost layer. Additionally, two depressions were excavated, lined with clay to form permanent pools, and planted with native vegetation. Eventually, the intent is to divert storm water to the site from an adjoining bus garage parking lot and nearby roadways. During a storm, diverted rainfall will temporarily pool on the site and infiltrate to the subsurface through the non-clay lined portions of the green space. Figure 1 shows the approximate location of the site on western Long Island.

Experimental Methods

Three undisturbed soil columns were excavated by hand from the site in East New York, Brooklyn. An approximately 35 cm length of black polyethylene culvert was placed over the column and minimal expansion foam (commercially available “Great Stuff” polyurethane) was injected into the gap between the soil and culvert. The soil columns were of varying depth, 0.33 meters in diameter, and only comprised the organic peat layer. The columns were transported to Ithaca, New York to conduct the laboratory experiments. Column dimensions and soil characteristics are detailed in Table 1. More detailed soil property information will be determined for each individual column when all experiments have been completed and the columns can be dismantled.

Table 1: Soil Column Properties

	Column A	Column B	Column C
Depth, cm	33	30.5	38
Infiltration Capacity, cm/hr	230	230	134
Soil Texture	54% coarse sand, 30% fine sand, 16% silt & clay		

A schematic of the experimental set-up is shown in Figure 2. For most of the trials, peristaltic pumps were used to deliver water to each column at a controlled rate. However, as the pumps had a maximum capacity of approximately 1 L/ min (82 cm/hr), a rate less than the maximum infiltration capacity, water was poured directly onto the columns for trials in which surface ponding was desired (Trial 4). Percolate was collected using a passive lysimeter consisting of fiberglass wicks (Boll et al., 1992). The lysimeter has 21 separate collection wicks evenly spaced in a grid pattern across the column bottom, thus allowing sampling from specific regions of the column. To limit the influence of preferential edge flow effects, samples were collected as a composite from the 9 inner wicks.

A synthetic stormwater solution intended to replicate average concentrations of metals and nutrients found in urban stormwater was prepared as detailed in Davis et al. (2001) and shown in Table 2.

Table 2: Chemical Make-up of Synthetic Stormwater Solution

Pollutant	Chemical	Chemical Conc. (mg/L)	Metal Conc. (mg/L)
Copper	CuSO ₄	0.08	0.021
Lead	PbCl ₂	0.08	0.059
Zinc	ZnCl ₂	0.6	0.28

Four trials were carried out on each of the three columns. Hydraulic application rate, duration, and constituents of interest for each trial are summarized in Table 3.

Table 3: – Summary of Trial Set-ups

Trial	Hydraulic Application Rate, cm/hr	Duration, min.	Constituents of Interest
1	8	720	Cl ⁻ , Cu, Pb, Zn
2	82	60	Cu, Pb, Zn
3	41	60	Cu, Pb, Zn
4	~247	7-10	Cu, Pb, Zn

Trial 1 applied a synthetic stormwater solution as well as a chloride tracer at a concentration of approximately 16 ppm (including Cl⁻ from metal salts). For the first two hours, samples were taken every 5 minutes. Sampling decreased to an interval of 30 minutes for the remaining time. Chloride concentrations were measured using a digital chloridometer (Buchler Instruments). Chloride was assessed in order to establish representative Breakthrough Curves (BTCs) for a conserved, non-reactive substance passing through each column. A breakthrough curve plots the concentration of a selected parameter (in this case, chloride) at the column bottom. BTCs for chloride in which transport is wholly dependent on matrix flow closely follow the advective-dispersion equation. In contrast, the presence of preferential flow results in an instantaneous initial jump in percolate chloride concentration.

Additional trials only applied the synthetic stormwater solution. For all trials, the columns were initially wetted with distilled water until all collection wicks drained at an approximately uniform rate. Once the columns were wetted through, the application rate of distilled water was sustained at the respective steady state rate of the trial for approximately 15 minutes before seamlessly transitioning to the application of the synthetic stormwater solution. In trials 2 and 3, samples were taken during the steady state application of distilled water and 5, 15, 30, 45, and 60 minutes after the synthetic stormwater solution application was initiated. In Trial 4, water was only applied for approximately 7 minutes (Column C required approximately 10 minutes due to a lower hydraulic conductivity). Samples were taken 2 and 5 minutes after the sythetic stormwater solution was initiated and approximately 1 minutes after the application was stopped.

Before running any trials with the synthetic stormwater solution, an approximately 4 pH synthetic rain water solution intended to replicate acid rain was applied to the columns (Richards et al., 2000). Percolate was collected and analyzed to determine background levels of Cu, Pb, Zn, phosphorus, and nitrate. Background levels for the metals are shown in Table 4. The pH=4 results are based on an average of two samples apiece taken from columns A and C. The pH=7 results are based on a sample taken during Trial 3 as the columns were being wetted prior to the addition of the synthetic stormwater solution.

Table 4: Background Metal Concentrations in Soil Percolate

	Column A		Column B		Column C	
	pH=4	pH=7	pH=4	pH=7	pH=4	pH=7
Cu	0.038	0.0049	---	0.0089	0.0295	0.0073
Pb	ND	ND	---	ND	ND	ND
Zn	0.052	0.0538	---	0.0126	0.0397	0.0044

Background metal concentrations at a pH of 4 are approximately an order of magnitude higher than background levels at a pH of 7. This finding is generally consistent with research indicating that metals tend to be more mobile under acidic soil conditions (McBride, 1994).

There is some cause for concern that the determination of copper removal efficiencies may be confounded by the fact that at pH=4 background copper concentrations shown in Table 4 were the same order of magnitude as the synthetic stormwater solution as shown in Table 2. However background levels were an order of magnitude less at a pH of 7, the neutral condition at which the actual removal efficiencies were assessed.

Cu, Pb, and Zn concentrations were measured at the Cornell Fruit and Vegetable Lab using an Inductively Couple Plasma Spectrophotometer.

3. Results and Discussion

Consideration of Cu Data Reliability

Figure 3 presents Cu and Zn concentrations for each of the four trials for the three columns. Pb is not included as the percolate Pb concentration was always below the detectable limit.

Particularly for Cu, percolate concentrations varied throughout the trial, generally starting high and diminishing through the trial. Rather surprisingly, in Figure 3a., at a time of zero (prior to the application of the synthetic stormwater solution) the observed copper concentration was higher than any other point during the run.

A comparison of metal concentrations measured in the influent synthetic stormwater solution indicates a large degree of variability in Cu concentrations when the Cu should be nearly constant. The influent metal concentrations should be nearly uniform as the influent, while made in batches, is based on the dilution of the same standardized high concentration metal solution. Table 5 summarizes the apparent Cu concentrations in the influent synthetic stormwater solution. For both Cu and Pb, the standard deviation between influent concentrations for the four trails is nearly equivalent to the mean values. The variability is potentially due to inconsistencies in the chemical analysis.

Table 5: Comparison of Measured Influent Metal Concentrations

	Metal Salt Conc.	Intended Metal Conc.	Used in Summary Analysis	8 (cm/hr)	41 (cm/hr)	82 (cm/hr)	247 (cm/hr)	Trials 1 to 4 Mean	Trials 1 to 4 St. Dev.
Cu (ppm)	0.08	0.021	0.02	0.1118	0.0019	0.0133	0.02	0.0368	0.0506
Pb (ppm)	0.08	0.059	0.03	0.0219	0.0052	0	0.0099	0.0093	0.0094
Zn (ppm)	0.6	0.28	0.3	0.2166	0.2171	0.3546	0.1279	0.2291	0.0936

Relationship of Removal Efficiency to Flow Rate

As the primary objective of these experiments, Figure 4 shows removal efficiencies as compared to flow rate. As noted in the discussion of Figure 3a through 3h, apparent removal efficiency fluctuates throughout certain trials, particularly for Cu. Thus, the removal efficiencies expressed in the figures are the mean of efficiencies observed across the trial.

Pb is not shown since in all trials, the effluent concentration was below the detectable limit. Apparent within Figure 4, the removal of zinc remains relatively constant at approximately 90% for all columns and all flow rates. The removal efficiency of copper is much more variable, both among columns and at different flow rates. To see if another method of quantifying removal within a trial would more successfully explain the relation between flow and removal, flow rate was compared to removal efficiency after 18 L of cumulative flow had been applied during each trial. This alternate method did not reveal any improved relationship. No definite statement regarding the relationship between Cu and flow rate can be made based on the data. This lack of conclusive results is not surprising given the unusual and inconsistent variability in the Cu data.

Determination of Preferential Flow

Figure 5-a, b & c shows the BTCs for each of the three columns as well as a best-fit of the one-dimensional advective-dispersion equation solved for a semi-infinite boundary condition. For each of the three BTCs, a plateau forms at a concentration of approximately 0.8 mg/L almost instantaneously. After the plateau forms, the breakthrough curves closely follow the advective-dispersion equation. The magnitude in the initial instantaneous jump can be used to quantify the fraction of preferential flow in relation to the fraction of matrix flow.

The advective-dispersion was fit to the data by adjusting the average matrix velocity and the diffusivity. Despite being based on a best-fit, these adjusted values are physically realistic. The values for each column are listed in Table 6.

Table 6: Advective-Dispersion Equation Fitting Parameters

	Column A	Column B	Column C
Pore Velocity, cm/min	0.9	0.9	1
Diffusivity, cm ² /min	8	10	7

If downward flow was uniform across the column cross-section and no preferential flow occurred, the expected velocity would be 0.28 cm/min. The diffusivities (D) are two orders of magnitude less than that determined for a soil column composed of 95% sand in which D equaled 0.076 cm²/min and V equaled 0.18 cm/min (Zurmuhl, 1998). However, D values are typically approximately proportional to the vertical pore velocity, placing the observed Ds in a reasonable range.

There is the possibility that the travel time of solute through the wick may influence the apparent breakthrough curve. A previous investigation into travel time through the fiberglass wick found that a chloride pulse moved at approximately 20.7 cm/min at a hydraulic application rate equivalent to approximately 9 cm/hr (Rimmer et al., 1994). Given that the wicks in our experiment are approximately 35 cm and the hydraulic loading rate is the same order of magnitude, the solute pulse should only be delayed approximately 100 seconds, relatively small compared to the sampling intervals we used.

In general, the presence of preferential flow indicates that 100% pollutant removal will not be possible as a certain fraction of inflow always bypasses the soil matrix structure. Based on the immediate measurement of approximately 1 mg/L chloride of a 15 mg/L influent solution, approximately 7% of the influent moves through the column by way of preferential flow. Thus, the presence of preferential flow provides some explanation of why removal efficiency is not strongly related to flow rate.

Variability Between Columns

Despite being a constructed system, the infiltration basin still makes use of natural materials that grow and are modified over time. Therefore due to the lack of homogeneity in root densities, earthworm activity, frost heave, etc. one would expect inherent spatial variability within the natural system. Three soil columns were used in order to try to capture some of this potential variability. Table 7 ranks columns within each trial based on the average observed metal concentration in the percolate. Again, Pb was not included as all Pb concentrations in the percolate were below the detectable limit.

Table 7. Ranking of Columns by Observed Metal Percolate Concentration

	8 (cm/hr)		41 (cm/hr)		82 (cm/hr)		247 (cm/hr)	
	Cu	Zn	Cu	Zn	Cu	Zn	Cu	Zn
Column A	Lo	Hi	Med	Med	Lo	Med	Med	Med
Column B	Hi	Med	Hi	Med	Hi	Med	Hi	Hi
Column C	Med	Lo	Lo	Lo	Med	Lo	Lo	Lo

Column B consistently had the highest percolate copper concentrations and alternated with Column A in having the highest zinc percolate concentrations. Column C consistently had the lowest observed percolate metal concentrations. This suggests that there are fundamental properties within the columns that affect the metal removal efficiency. As the most obvious factor, Column C was found to have the lowest hydraulic conductivity as assessed during Trial 4.

4. Conclusions and Recommendations

A series of laboratory experiments were conducted to evaluate the pollutant removal capacity of an organic matter layer overlaying a highly conductive sublayer. Of primary interest was to assess whether the organic matter layer can remove pollutants at the high hydraulic loading rates permitted by the relatively high conductivity of the substrate. Experiments showed that the removal efficiencies of Pb, Cu, and Zn were independent of hydraulic application rate. Removal of Pb and Zn was consistently above 90% while removal of Cu was more variable but was generally near 50%.

Breakthrough curves illustrated that approximately 10% of the flow through the columns travels through macropores, not matrix flow. Due to the rapid transport through macropores, little pollutant removal is likely to occur. Thus, any organic matter layer applied under natural conditions, will be unlikely to achieve a removal efficiency much greater than 90%.

Several recommendations can be made for future experiments. First, the experiments could be repeated using higher metal concentrations in the synthetic stormwater solution, even if these concentrations do not match levels routinely observed in actual stormwater. Primarily, these higher levels would minimize the significance of background metal levels. Additionally, the experiments could be repeated using a synthetic stormwater solution decreased to a pH typical of acid rain. One presumes that metals would be more mobile at lower pH levels.

A more extensive series of experiments could consider long-term removal for a more complete hydrologic cycle in which there are periods of wetting and drying. In particular, metal concentrations would be measured as the columns are wetted with pure distilled water void of any metals. During this wetting, one would anticipate desorption of metals since influent concentrations would not be in equilibrium with soil metal levels in the column.

A final suggestion is to run the experiments using pesticides and organic compounds instead of metals, as these compounds are more commonly found in groundwater than trace metals.

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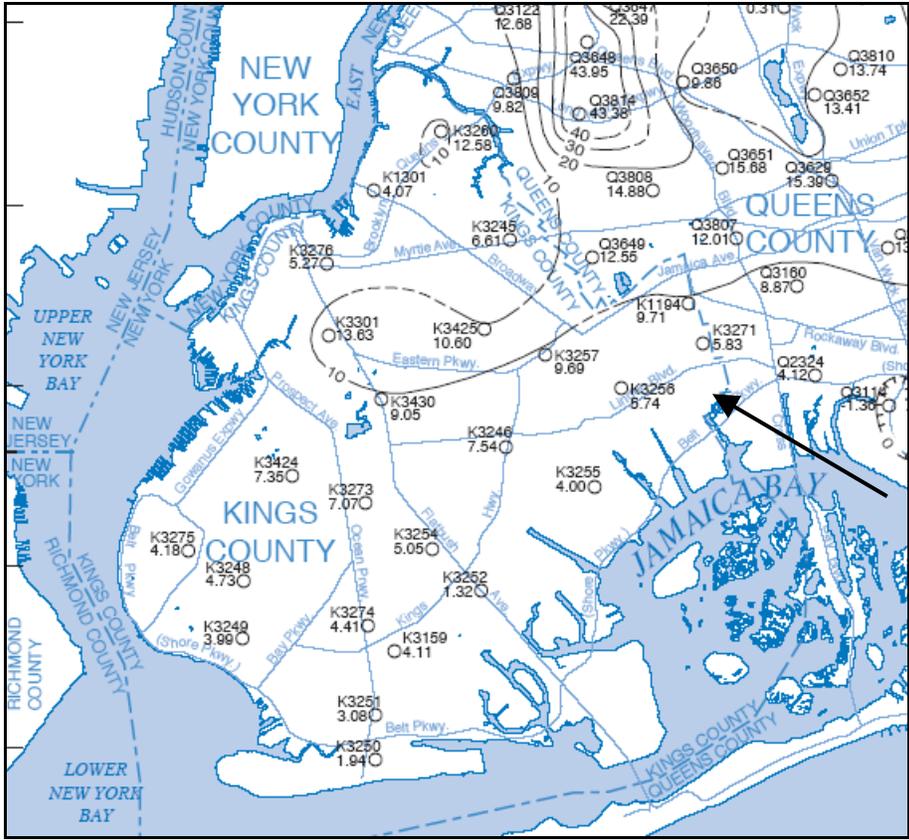


Figure 1. Site location shown on USGS map of the Upper Glacial Aquifer on Western Long Island, March-April 2000 (source: Ron Busciolano, USGS WRIR 01-4165)

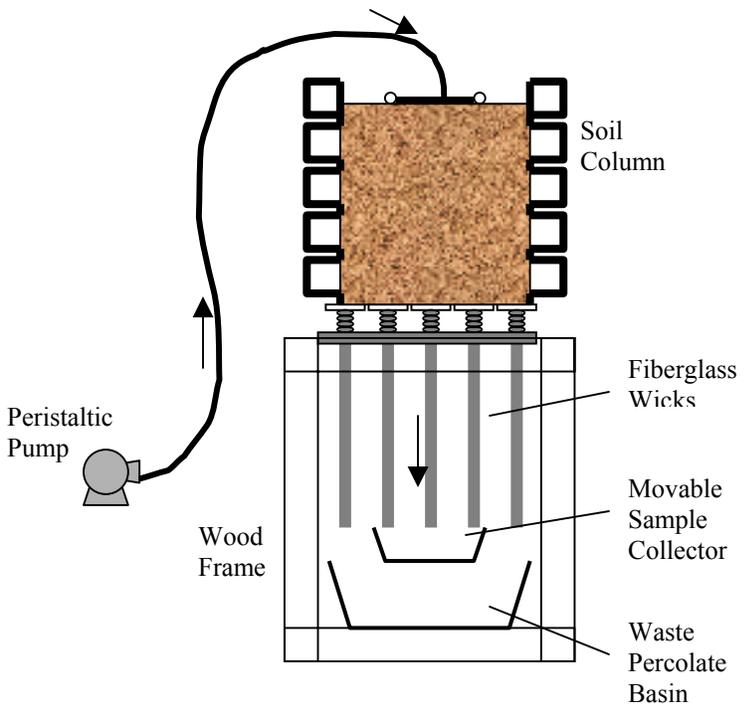


Figure 2. Schematic of experiment set-up.

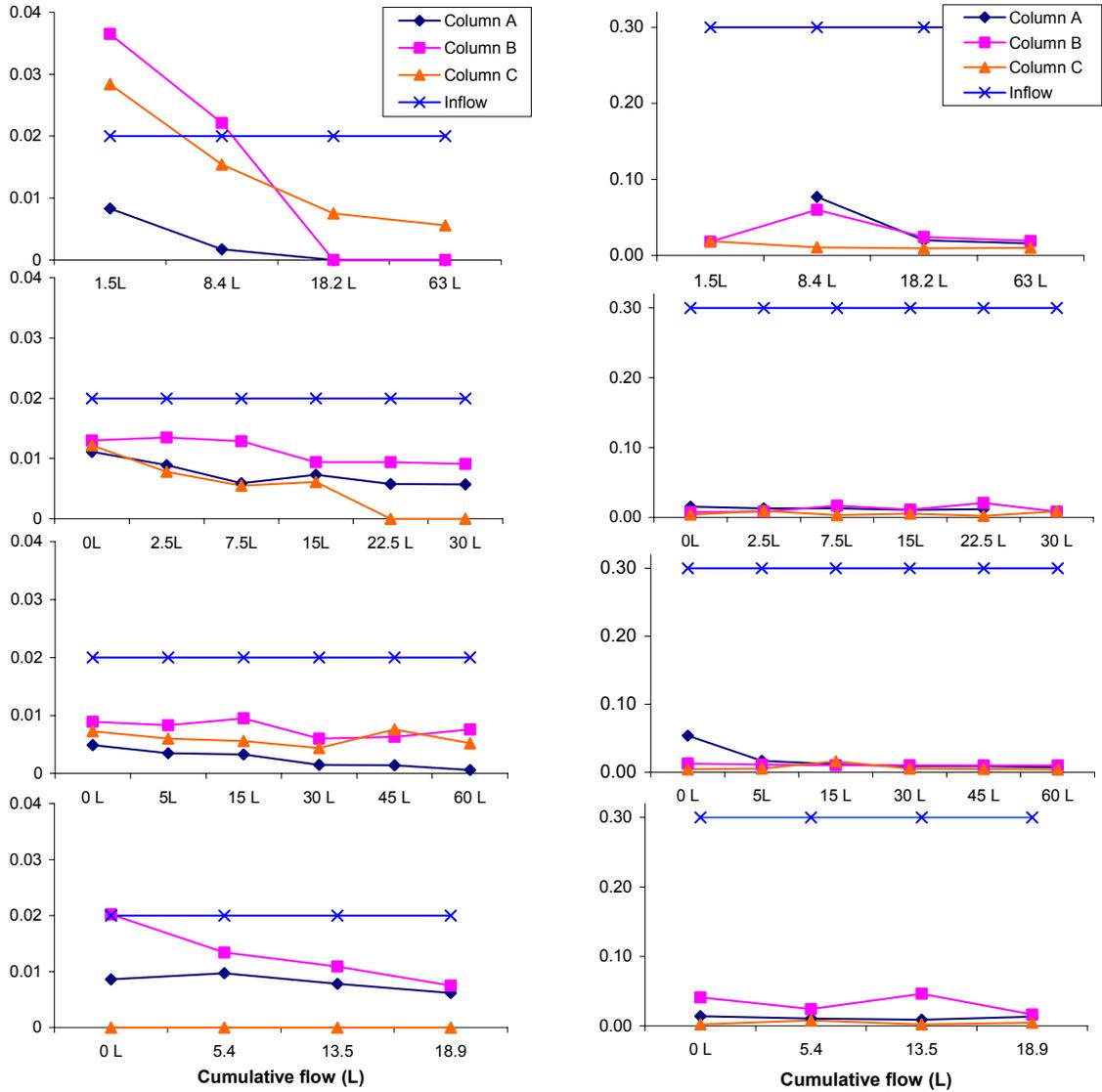


Figure 3: Column Percolate Concentrations of Copper and Zinc at cumulative flows. Graphs A through D represent Copper concentrations at 8 cm/hr, 42 cm/hr, 84 cm/hr and 247 cm/hr respectively. Graphs E through H represent Zinc concentrations at 8 cm/hr, 42 cm/hr, 84 cm/hr and 247 cm/hr respectively. Data included for the three different soil columns: diamond – column A, square – column B, triangle – column C, and x – inflow concentration.

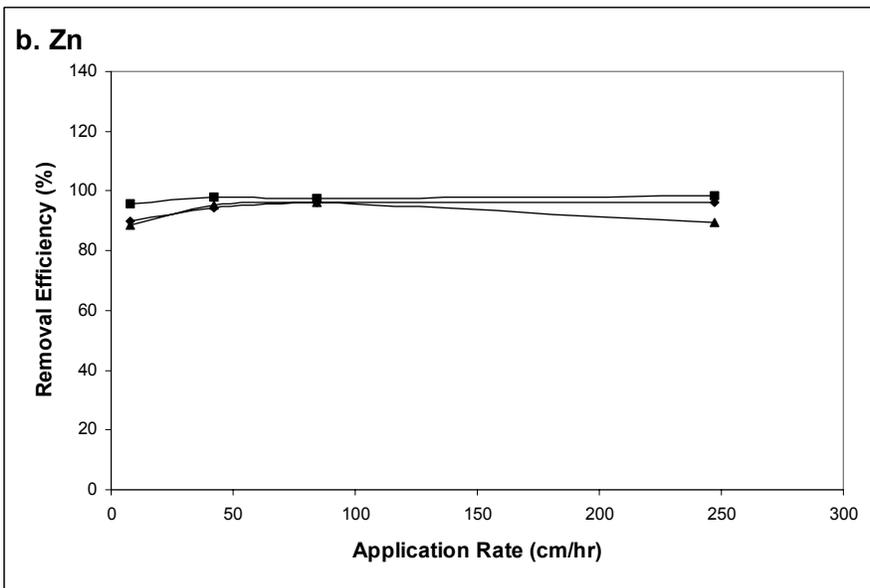
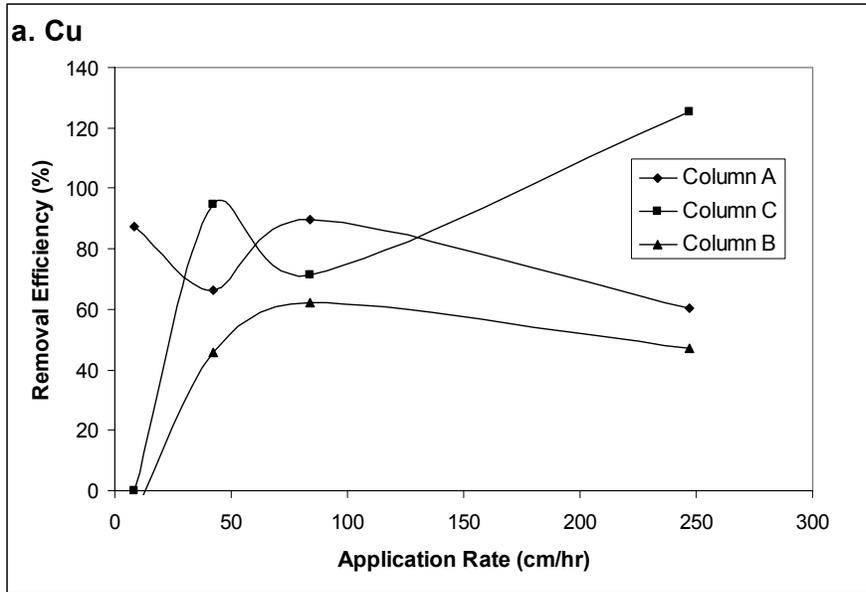


Figure 4. Metal removal efficiencies for varying hydraulic application rates on three different soil columns: Diamond - Column A, Triangle - Column B, Square - Column C. Pb is not shown as removal was consistently below detection limit.

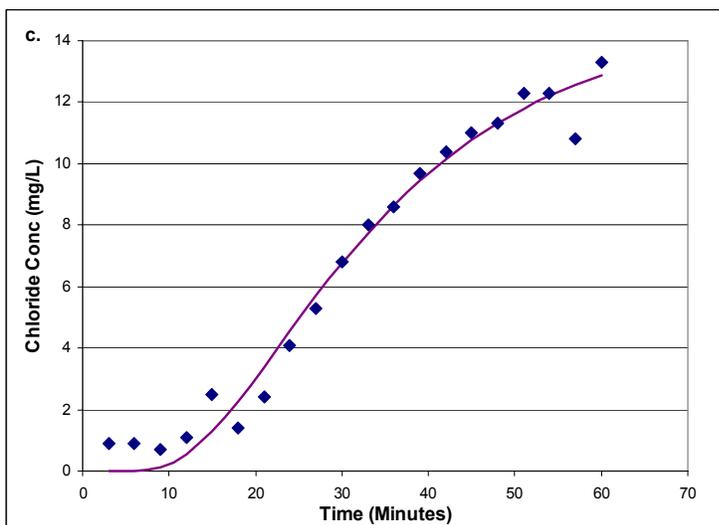
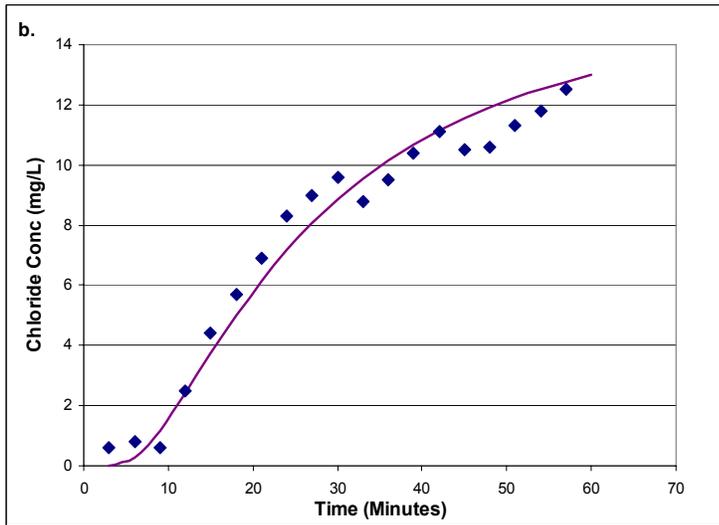
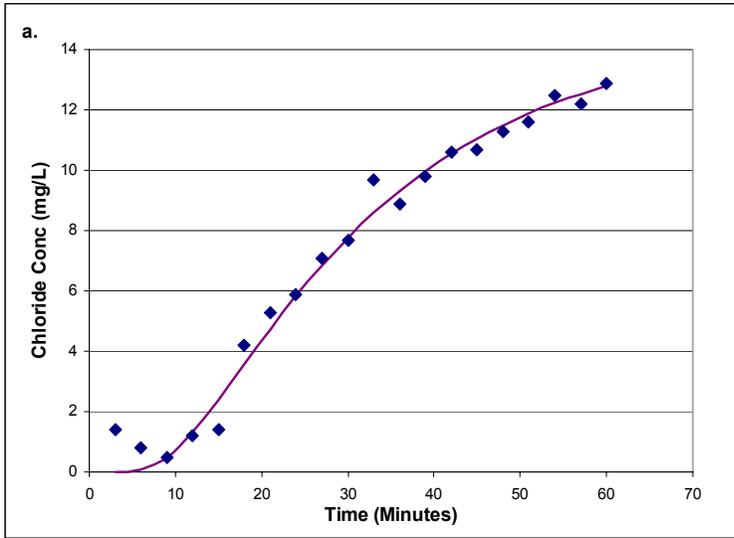


Figure 5. Observed chloride concentrations (points) versus chloride concentrations predicted using the advective-dispersion equation for columns a, b, and c. The plateau in the observed data in the first several minutes is indicative of preferential flow.

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References:

Ayers, M. A., J.G. Kennen, and P.E. Stackelberg. 2000. Water Quality in the Long Island and New Jersey Coastal Drainages, 1996-1998. USGS Circular 1201. *on-line at* <http://pubs.water.usgs.gov/circ1201/>

Boll, J., T.S. Steenhuis, and J.S. Selker. 1992. Fiberglass Wicks for Sampling of Water and Solutes in the Vadose Zone. *Soil Science Society of America Journal*. Vol. 56, No. 3, pp 701-707.

Clark, S. and R. Pitt. 1999 Stormwater Treatment at Critical Areas – Evaluation of Filtration Media. National Risk Management Research Lab. US EPA. EPA/600/R-00/010.

Committee on Ground Water Recharge, Water Science and Technology Board, Commission on Geosciences, Environment, and Resources. 1994. Ground Water Recharge Using Waters of Impaired Quality. National Academy Press. Washington, DC. *Online at* <http://books.nap.edu>

CSF Treatment Systems Inc. Technical memorandum – Three Year Performance Study – 185th Avenue. Prepared by CSF Treatment Systems Inc. Portland, Or, 1994.

Davis, A. P., M. Shokouhian, H. Sharma, and C. Minami. 2001. Laboratory Study of Biological Retention for Urban Stormwater Management. *Water Environment Research*. Vol. 73, No. 1, pp 5-14.

Environmental Protection Agency. 1983. Results of the Nationwide Urban Runoff Program. PB 84-185552. US EPA Water Planning Division. Washington, DC.

Farm, C. 2002. Metal Sorption to Natural Filter Substrates for Storm Water Treatment – Column Studies. Vol. 298, pp 17-24.

McBride, M.B. 1994. *Environmental Chemistry of Soils*. Oxford University Press, New York.

Pitt, R., R. Field, M. Lalor, M. Brown. 1995. Urban Stormwater Toxic Pollutants: Assessment, Sources, and Treatability. *Water Environment Research*. Vol. 67, No. 3, pp 260- 275.

Richards, B.K., T.S. Steenhuis, J.H. Peverly, and M.B. McBride. 2000. Effect of sludge-processing mode, soil texture and soil pH on metal mobility in undisturbed soil columns under accelerate loading. *Environmental Pollution*. 109: 327-346.

Rimmer, A., T. S. Steenhuis, J. S. Selker, and G. Albrecht. 1994. Wick Samplers: An Evaluation of Solute Travel Times. *Soil Science*. 159(4):235-243.

Sansalone, J.J. 1999. Adsorptive Infiltration of Metals in Urban Drainage – Media Characteristics. *Science of the Total Environment*. Vol. 235, pp 179-188.

Seaburn, G.E. and D.A. Aronson. 1974. Influence of Recharge Basins on the Hydrology of Nassau and Suffolk Counties, New York. USGS Water Supply Paper 2031.

Zurmuhl, T. 1998. Capability of Convective Dispersion Transport Models to Predict Transient Water and Solute Movement in Undisturbed Soil Columns. *Journal of Contaminant Hydrology*. Vol. 30, pp 101-128.